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Viewpoint

Microtidal estuaries warrant special management measures that recognise their critical vulnerability to pollution and climate change

Richard M. Warwick[a,b,c,*], James R. Tweedley[b,c], Ian C. Potter[b,c]

[a] Plymouth Marine Laboratory, Prospect Place, West Hoe, Plymouth PLI 3DH, UK
[b] Centre for Sustainable Aquatic Ecosystems, Harry Butler Institute, Murdoch University, 90 South St, Murdoch, Western Australia 6150, Australia
[c] School of Veterinary and Life Sciences, Murdoch University, 90 South Street, Murdoch, Western Australia 6150, Australia

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ABSTRACT

Not all estuaries are equally susceptible to anthropogenic perturbation. Microtidal estuaries with long residence times are intrinsically less robust than well-flushed macrotidal estuaries, facilitating the accumulation of contaminants. This promotes development of blooms of non-toxic and toxic phytoplankton, and hypoxia and anoxia may occur in deeper sections of the typically stratified water column. In Mediterranean and arid climates, high temperatures and low and/or seasonal rainfall can result in marked hypersalinity. Thus, any increase in anthropogenic perturbation will further decrease the health of a system in which the biota already experiences natural stress. Microtidal estuaries are also more susceptible to climate change, the detrimental longer-term effects of which are becoming manifestly obvious. Numerous attempts have been made to develop novel solutions to problems caused by eutrophication, phytoplankton blooms, hypoxia and hypersalinity, which have met with various levels of success, but the need for such measures and effective legislation is increasingly critical.

1. Introduction

Estuaries are highly productive and ecologically important from an ecosystem services perspective and in particular for fisheries (Schelske and Odum, 1961; Potter et al., 2015). Yet, temperate estuaries are regarded as the most degraded of all marine ecosystems (Jackson et al., 2001; Kennish, 2002). Most of this degradation has resulted from anthropogenic effects that have arisen as a direct consequence of human colonization and development of the estuary and its catchment, the production and release of human and industrial waste, some of which is toxic, and from nutrient run-off from urban and agricultural land use (Wilson, 1988; McLusky and Elliott, 2004; Tweedley et al., 2015). Not all estuaries are equally susceptible to perturbation, however, and we argue below that the extent of perturbation largely depends on the tidal regime and is exacerbated by climate.

In this essay, we adhere to the following definition of an estuary as given in Potter et al. (2010). "An estuary is a partially enclosed coastal body of water that is either permanently or periodically open to the sea and which receives at least periodic discharge from a river(s), and thus, while its salinity is typically less than that of natural sea water and varies temporally and along its length, it can become hypersaline in regions when evaporative water loss is high and freshwater and tidal inputs are negligible". Microtidal estuaries, *i.e.* those with a tidal range < 2 m, in Mediterranean climates generally have narrow mouths, are poorly flushed and have relatively long water residence times, *i.e.* typically weeks to months, whereas macrotidal estuaries are generally funnel-shaped and well-flushed, with relatively short residence times, *i.e.* typically hours to days (Uncles et al., 2002; Tweedley et al., 2016b). For the purposes of this essay, the term macrotidal estuary includes mesotidal estuaries with a tidal range of 2–4 m and those with greater tidal ranges. As we have published elsewhere a detailed review of the contrasting features of the ecology of macrotidal and microtidal estuaries (Tweedley et al., 2016b), we allude only briefly to those features that are directly relevant to environmental management in order to produce a concise and readable Viewpoint and avoid unacceptable repetition.

In some microtidal estuaries, highly seasonal and/or low rainfall results in a sand bar forming at the mouth of the estuary, which prevents the exchange of water with the ocean and further increases the residence time (Ranasinghe and Pattiaratchi, 1998; Chuwen et al., 2009). Estuaries can become closed to the ocean either intermittently, seasonally or normally, and in the last case the residence time is infinite for the years of closure. These estuaries that become closed from the ocean for periods are numerically dominant in microtidal Mediterranean climates and constitute 39 (82%) of the 47 estuaries in south-western Australia (Tweedley et al., 2017b) and 75% of the total number

* Corresponding author.
E-mail address: rmw@pml.ac.uk (R.M. Warwick).

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of ~250 estuaries in southern Africa (Whitfield, 1998).

2. Susceptibility to pollution

Estuaries with long residence times are intrinsically less robust than better flushed estuaries, because they facilitate the accumulation of contaminants. Furthermore, even when there are no anthropogenic inputs, naturally-occurring dissolved and particulate organic material from terrestrial sources accumulate in these systems and this can lead to a marked depletion in dissolved oxygen (Nixon et al., 1996; Wolanski, 2007).

This problem is exacerbated in the microtidal estuaries of south-western Australia because the residence times are longest in the warmer months of the year, when rainfall is very low, and will be particularly marked in those systems in which the mouth closes and thereby prevents flushing. For example, the residence time in the upper reaches of the permanently-open Swan-Canning Estuary is estimated to be 235 days in summer but only 1 day in winter (Kalnejais et al., 1999). High temperatures increase the rates at which particulate organic matter becomes decomposed and dissolved nutrients are utilized, sometimes giving rise to excessive blooms of phytoplankton (Lukatelich and McComb, 1986; Robson and Hamilton, 2004). In a climate with less seasonal rainfall, such as in Denmark, microtidal estuaries become stratified during periods of low rainfall, have high nutrient concentrations and feature large plankton blooms, hypoxia and anoxia (Conley et al., 2000). However, these estuaries are fjordic, with hard bottoms that support high densities of filter-feeding macrobenthos, such as the bivalve *Mytilus* and ascidian *Ciona*, that can greatly reduce phytoplankton concentrations and ameliorate these effects to some degree (Conley et al., 2000).

In contrast, macrotidal estuaries are usually very turbid because strong tidal scour re-suspends bottom sediments, producing local accumulations of suspended particles at the turbidity maximum which, together with riverine inputs of terrestrial material and irrespective of nutrient concentrations, inhibits light penetration so that primary production is strongly light-limited, for example in the Schelde (Soetaert and Herman, 1994), Gironde (Irigoien and Castel, 1997) and Severn estuaries (Underwood, 2010; Kadiri et al., 2014). In these estuaries, only a small proportion of the nutrients supplied by the rivers is used by the phytoplankton and the incidence of toxic algal blooms is minimal. On the other hand, a large proportion of the nutrients is utilized by the phytoplankton in microtidal estuaries, where low turbidity results in clear water, making these systems far more sensitive to increases in nutrient input that might lead to excessively high phytoplankton biomass (Monbet, 1992).

Elevated concentrations of dinoflagellates and cyanobacteria are characteristic of these blooms. They are symptomatic of eutrophication in microtidal estuaries and may result in environmental problems since many species in these two taxa are toxic and mainly responsible for the harmful algal blooms that frequently occur in microtidal estuaries throughout the world (Hall et al., 2008; Place et al., 2012). These blooms have serious consequences, including large fish kills, as for example in the Peel-Harvey and Swan-Canning estuaries in south-western Australia (Potter et al., 1983; Lenanton et al., 1985; Hallett et al., 2016), and an increased danger to human health through diarrhetic and paralytic shellfish poisoning, as for example in Alfacs Bay on the Mediterranean coast of Spain (Artigas et al., 2014). Many estuaries with the greatest annual phytoplankton production are microtidal, such as Chesapeake Bay on the eastern coast of the USA and the Swan-Canning Estuary (Cloern et al., 2014). Within a single microtidal estuary, short-term variation in phytoplankton production may be related to variations in residence time, for example in the Patos Lagoon Estuary in southern Brazil (Odebrecht et al., 2005).

While the water column in macrotidal estuaries is well- or at least partially-mixed, driven by tidal action, microtidal estuaries are typically highly stratified, with little or no mixing between the fresh water on the surface and the salt water below, resulting in the production of a halocline (Dyer, 1997). Strong salinity stratification encourages high rates of sediment deposition (Traykovski et al., 2004; Ralston et al., 2012), thus enhancing nutrient recycling (Hopkinson et al., 1999; Watanabe et al., 2014) and which can lead to conditions that produce hypoxia (< 2 mg oxygen L^{-1}) and even anoxia (Paerl et al., 1998; Kurup and Hamilton, 2002; Tweedley et al., 2016a). Vertical mixing in macrotidal estuaries tends to prevent hypoxia, which is thus not generally a problem in these systems (Lanoux et al., 2013). Although cycles of sediment erosion and deposition occur in macrotidal estuaries on daily, weekly, lunar, equinoctial and seasonal cycles (Wolanski and Elliott, 2016), posing a physical stress not present in microtidal estuaries, these events have been operating over evolutionary time scales so that the biota have become well adapted to them. This ability to absorb stress without adverse effects has been termed “environmental homeostasis” by Elliott and Quintino (2007) and estuarine organisms may regard this environmental stress as a subsidy whereby they successfully capitalise on such conditions, which may only be regarded as stressful for marine or freshwater-adapted organisms.

Naturally high levels of organic matter and oxygen depletion will encourage the success of families of benthic macroinvertebrates (*e.g.* Capitellidae, Spionidae and Chironomidae) that are characterized by high AMBI scores and even occur in the essentially-pristine Broke Inlet in south-western Australia (Tweedley et al., 2014, 2017b). AMBI (the AZTI marine biotic index) is based on scoring benthic macroinvertebrate species according to their known sensitivities to environmental perturbations, thus enabling an average score to be obtained for the assemblage as a whole (Borja et al., 2000). Such high AMBI scores elsewhere would be associated with pollution produced by anthropogenic sources of organic matter (Borja et al., 2000). Thus, any increase in anthropogenic perturbation will further decrease the health of a system in which the biota is already under stress from natural causes.

3. Susceptibility to climate change

The nature of climate change and its effects on the macrotidal estuaries of the northern hemisphere are variable, although none of the predictions imply catastrophic environmental consequences (Robins et al., 2016). Microtidal estuaries are likely to be more susceptible to climate change, the longer-term effects of which are obvious and already manifesting themselves, while their causes are not being adequately addressed at a global scale. The estuaries of south-western Australia, for example, have been subjected to sustained warming and drying. Annual mean air temperatures in this region rose by 1.1 °C between 1910 and 2013 and, by 2030, are predicted to be 0.5–1.1 °C above the 1986–2005 average, with respective increases of 1.2–2.0 °C and 2.6–4.0 °C by 2090 under moderate (Representative Concentration Pathway; RCP4.5) and high (RCP8.5) emission scenarios (Hope et al., 2015; Hallett et al., 2018). Representative concentration pathways (RCPs) have been developed for the climate modelling community as a basis for long-term and near-term modelling experiments and are referred to as pathways in order to emphasize that their primary purpose is to provide time-dependent projections of atmospheric greenhouse gas concentrations (van Vuuren et al., 2011).

The ~15% decline in rainfall, which has occurred in south-western Australia since the mid-1970s, has led to a 70% reduction in runoff into reservoirs (Petrone et al., 2010; McFarlane et al., 2012). Similarly, the average annual freshwater flow into the Coorong in South Australia has been reduced by about 70% over recent decades and is highly variable, with no flow over the barrages that have been built across the estuary for several consecutive years during a “Millennium Drought” between 1997 and 2009 (Kingsford et al., 2011; Leblanc et al., 2012). Decreased river flows resulted in an increase in residence time and the frequency of mouth closure, thereby producing stressful hypoxic and hypersaline (> 40) conditions, with the maximum recorded salinity approaching 200 (Wedderburn et al., 2016). Other examples of extreme

hypersalinity in microtidal estuaries include Lake St Lucia in South Africa and the Wellstead, Hamersley, Stokes and Culham inlets in south-western Australia (Young and Potter, 2002; Whitfield et al., 2006; Chuwen et al., 2009). The last of those estuaries provides an extreme example of this type of estuary, with salinities reaching as high as 312 when the mouth of the estuary had not opened for at least three years (Hoeksema et al., 2018).

As the extent of hypersalinity in the Laguna Tamaulipas (USA/Mexico) increased the number of fish species declined (Copeland, 1967), as also occurred in the lower part of the Wellstead Estuary (Young and Potter, 2002). In the latter estuary and the Coorong, the only species that was still caught when salinities reaches ~120 were atherinids, *i.e. Atherinosoma elonagta* and *Atherinosoma microstoma*, respectively (Wedderburn et al., 2016). As salinities rose above 50 in Lake St Lucia, all benthic macroinvertebrates species except chironomid larvae and ostracods were no longer found (Boltt, 1975). Furthermore, as the traditional molluscan and crustacean prey of the more salinity-tolerant fish species, *i.e.* the haemulid *Pomadasys commersonnii* and sparid *Rhabdosargus sarba,* were no longer available, these species shifted to feeding on filamentous algae (Wallace, 1975; Tweedley et al., 2018).

Mortalities of > 6000 fish, representing at least 11 species, were recorded in the Seekoei Estuary in South Africa as salinities rose above 90 during a period when this system was closed (Whitfield, 1999). Moreover, the rise in salinities in Culham Inlet to more than twice that of full strength seawater led to the death of ~1.3 million individuals of the sparid *Acanthopagrus butcheri* (Hoeksema et al., 2006). The blooms of dinoflagellates and cyanobacteria that occur under hypersaline conditions in St Lucia also result in fish kills (Grindley and Heydorn, 1970; Muir and Perissinotto, 2011). Such problems do not occur in typical macrotidal estuaries as river flows are less important than tidal exchange, and they never become hypersaline. Motile zooplankton and fish assemblages in microtidal estuaries comprise several abundant autochthonous species, *i.e.* undergoing their whole life in these systems, whereas those in macrotidal estuaries consist of allochthonous species, *i.e.* those that do not originate in the estuary but are advected or migrate into the estuary from either the river or the sea (Tweedley et al., 2016b; Plenty et al., 2018). Consequently, long-term reductions in river flow due to climate change are relatively more important for these taxa in microtidal estuaries. For diadromous and marine estuarine-opportunist fish species, reduced estuarine outflows may result in reduced cues associated with estuary mouths, and they will be precluded altogether by mouth closure (Gillanders et al., 2011).

Sea level rise is another feature of climate change that will potentially impact on estuaries. The sea level in south-western Australia (measured at Fremantle) has risen by an average of 1.4 mm per year from 1966 to 2009 and, under RCP4.5, is predicted to increase by 7–17 cm by 2030 and 28–66 cm by 2090 (Hope et al., 2015). Rising sea levels will increase the susceptibility of estuaries in this region to coastal flooding associated with extreme sea level and storm surge events (Hallett et al., 2018). However, an increased frequency of overtopping of sand bars and thus an increased input of marine water could have an ameliorating effect when there is pronounced hypersalinity.

Climate change impacts on microtidal estuaries will be exacerbated by their synergistic effects in combination with the anthropogenic and natural environmental stresses to which they are currently subjected. For example, many pollutants will exhibit increased toxicity at higher temperatures (Ficke et al., 2007), and decreasing rainfall will increase the concentration of nutrients arising from agricultural runoff. The spawning thresholds of many fish and invertebrate species will need to adapt to reduced water flows and an increase in salinity and temperature. However, the resultant combined effects of these physico-chemical variables on the physiology of species in microtidal estuaries have yet to be elucidated.

4. Management implications

The natural environmental stresses to which microtidal estuaries are subjected become exacerbated by anthropogenic influences, while the lack of natural stress beyond the adaptive capabilities of the species present is the baseline condition in macrotidal estuaries (Elliott and Quintino, 2007). Although the latter estuaries may of course be in poor environmental condition, this is generally the result of historical neglect and pollution. In such circumstances several estuaries, such as the Delaware Estuary (USA) and Thames Estuary (UK), have been successfully remediated as a result of effective legislation, pollution control and the setting of water quality objectives (McLusky and Elliott, 2004). Such macrotidal systems have a greater recovery potential since their high rates of tidal flushing enable them to respond relatively rapidly to reductions in the input of domestic and industrial effluents. In microtidal estuaries, contaminated fine particulate material may be deposited in the deeper waters where current velocity is typically low and the increase in salinity aids in flocculation and deposition (Ryan et al., 2003). For example, up to 80% of the fine sediments transported into the estuary become 'trapped' there (Patchineelam et al., 1999, Roy et al., 2001), constituting a very effective sink for this material.

The absence of pronounced flushing in microtidal estuaries reduces the recovery potential of these systems following the kinds of management measures noted above for macrotidal estuaries. The fragility of microtidal estuaries thus warrants management measures that recognise their special environmental characteristics and this poses difficult problems for managers and policy makers. Numerous attempts have been made to develop novel solutions to problems caused by eutrophication, phytoplankton blooms, hypoxia and hypersalinity. Attempts to control excessive phytoplankton production have been made in several estuaries, including Chesapeake Bay, through the restoration of oyster reefs by providing hard substrata and seeding with wild-caught or cultured spat (Coen et al., 2007). The primary purpose of such restorations and their effectiveness in phytoplankton control are still under debate, and limited success in the case of Chesapeake Bay is perhaps a problem of scale. To control phytoplankton production in such a large area (11,601 km^2) is a daunting task, whereas the largest microtidal estuary in south-western Australia is only 130 km^2, with a similar situation on the temperate southern coast of South Africa. Suspension feeders have been shown to increase denitrification and enhance nutrient sequestration, thus reducing the effect of eutrophication (Kellogg et al., 2013) and this may also help to alleviate the predicted effects of increased levels of hypoxia (Najjar et al., 2010). There is potential for other filter-feeding species such as mussels to fulfil this role (Dame, 1993; Dame and Prins, 1997; Wilcox et al., 2018).

Farming practices also need to be strictly controlled to prevent excessive nutrients entering the estuaries. This includes fertilizer management (*e.g.* testing of soil to ensure that fertilizer is only used when actually needed), provision of adequate animal waste treatment and construction and maintenance of fences to prevent animals from trampling on riparian vegetation and defecating in the rivers.

Attempts to break down stratification and therefore prevent the development of hypoxia and anoxia in deeper waters of the upper Swan-Canning Estuary in south-western Australia using bubble plumes created by pumping compressed air through a porous pipe have not been successful (Hamilton et al., 2001). However, artificial oxygenation has been used effectively to alleviate stratification-induced hypoxia. Currently, four oxygenation plants have been constructed on the banks of narrow upstream reaches of the Swan-Canning Estuary. These fixed plants pump oxygen-depleted water from near the riverbed, supersaturate it with oxygen and return the newly oxygenated water to the bottom waters of the estuary through several micro-diffusers (Hipsey et al., 2013). Although this method prevents water column hypoxia (Huang et al., 2018), oxygen was only found to penetrate between 1 and 1.5 mm into the sediment (Norlem et al., 2013). As these plants rely on the small tidal water movements of the salt wedge to oxygenate a

Fig. 1. Examples of environmental degradation and remediation measures in the Vasse-Wonnerup Estuary, south-western Australia. Namely, blooms of (a,b) phytoplankton and (c,d) macroalgae; (e,f) a large fish kill; (g) monosulfidic black ooze exposed by declining water levels and (h) anaerobic decomposition of organic material resulting in the release of hydrogen sulphide from the sediment; (i) drying of the estuary leading to hypersalinity (salinity = 40) to (j) only pools remaining (salinity = 132) and eventually (k) complete drying. (l) Opening of the bar to allow exchange of water with the ocean and (m) an oxygenation plant used to prevent hypoxia and anoxia. Photographs e, f and l supplied by the Department of Water and Environmental Regulation.

wider area than in the immediate vicinity of the diffusers, the effects of each plant are spatially limited, *i.e.* two plants in the Swan axis of the Swan-Canning Estuary are estimated to oxygenate an area of between 0.6 and 1.2 km^2, which equates to ~1.2–2.4 km in horizontal stream distance (Hipsey et al., 2013). When hypoxic events are spatially and temporally variable, oxygenation plants constructed on vessels such as the 'Thames Bubbler' could be employed to ameliorate small-scale events, and also potentially create oxygen refugia if hypoxia occurred over a wider area (Lloyd and Whiteland, 1990).

Major engineering works to deepen channels and manage bar openings may also improve the quality of microtidal estuaries by improving exchange of water with the ocean. Dredging has been used effectively to increase water circulation in the hypersaline Laguna Madre in the USA/Mexico and reduced the proportion of salinity values > 50 from 33–38% to only 2.6–2.8% (Quammen and Onuf, 1993). Extreme eutrophication in the microtidal Peel–Harvey Estuary in south-western Australia was manifested by extensive growths of macroalgae and the development of prolific seasonal blooms of the cyanobacterium *Nodularia spumigena* (McComb et al., 1981), which led to fish kills (see above). To ameliorate these effects a second artificial entrance channel, the Dawesville Cut (2.5 km long, 0.2 km wide and 4–6.5 m deep), was constructed. This led to a tripling in tidal water exchange with the

ocean, a 2–4 fold decrease in total phosphorous and nitrogen concentrations and a flushing of 10% of the estuary volume each day (Ruibal-Conti, 2014; Elliott et al., 2016). Moreover, the resultant marked increase in salinity produced conditions far less conducive to the growth of *N. spumigena,* preventing the regular occurrences of blooms (Potter et al., 2016).

Artificial opening of the bars that close the mouths of many microtidal estuaries is employed in some systems in both South Africa and Australia to allow exchange of water with the ocean and thus reduce hypersalinity, prevent hypoxia and increase water levels so as to maintain feeding areas for birds (Branch et al., 1985; Tweedley et al., 2017a). This management action is used in the Vasse-Wonnerup, a small microtidal estuary in south-western Australia, which suffers from severe eutrophication, hypoxia, toxic algal blooms, drying, hypersalinity, fish kills and poor sediment quality, including the presence of monosulfidic black ooze (Fig. 1; Tweedley et al., 2017a).

The ecological effects of declining freshwater flows under a drying climate will be aggravated by increasing water extraction for human use (Vörösmarty et al., 2000). In South Australia, for example, upstream diversion of water in the Murray-Darling Basin magnifies the effects of drought lower downstream in the estuary (Kingsford et al., 2011). In such circumstances, maintaining natural water flows and decreasing abstraction is an important approach to reducing vulnerability, particularly to the effects of drying and warming (Palmer et al., 2008). Increased salinity might be reduced by artificially increasing water flows, perhaps *via* cloud seeding, inflows from dam storages, or supplementation from artisanal bores (Hallett et al., 2018).

As microtidal estuaries are predisposed to natural environmental stress and differ markedly from their macrotidal counterparts in terms of their physical-chemical characteristics and biota, these systems require drastic and potentially expensive measures to prevent and mitigate the frequency and severity of deleterious events. Such measures include the kinds of major engineering works, changes to farming practices and development of biological control mechanisms noted above. Moreover, given the severe effects of climate change which, in Mediterranean climates, is likely to reduce freshwater flow and increase the occurrence and magnitude of stratification, hypoxia and hypersalinity, there is an urgent need to develop new management strategies. The effectiveness of these remediation measures should be evaluated by adequate real-time monitoring of both physico-chemical and biotic components of estuaries.

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